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FECUNDITY AND RECRUITMENT POTENTIAL OF COASTAL CUTTHROAT TROUT IN OREGON AND WASHINGTON

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Fecundity of coastal cutthroat trout, *Oncorhynchus clarki clarki*, is a critical life history parameter that has been reported only cursorily in the published literature. Regression equations were calculated for fecundity as related to fork length and weight for coastal cutthroat trout from Oregon and Washington. Fecundity was positively correlated to length and weight for females from Oregon and from Washington, although there was a wide range in fecundity for females of similar sizes. Estimated fecundity was significantly higher, albeit small, for Washington trout when compared with Oregon trout of the same length and weight. The average fecundity of Oregon trout was 802 eggs for fish with an average length of 364 mm (n = 90) and an average weight of 576 g (n = 84). Washington trout with an average length of 372 mm (n = 88) and an average weight of 661 g (n = 76) had an average fecundity of 1,108 eggs. Predicted fecundity from grouped data for Oregon trout increased from 121 eggs for females with a mean fork length of 218 mm to 1,278 eggs for 466 mm fish and from 421 eggs for 273 g females to 1,556 eggs for 1,152 g females. Predicted fecundity from grouped data for Washington trout increased from 452 eggs for 265 mm fish to 1,675 eggs for 483 mm fish and from 716 eggs in 234 g females to 2,240 eggs for 1,715 g females. Fecundity estimates were used to estimate egg deposition and recruitment potential for coastal cutthroat trout from three Oregon and two Washington streams for which the only reported data on upstream and downstream movement of trout was available. Estimated recruitment potential for these streams demonstrated that survival from egg deposition to smolt, smolt to first return to freshwater, second, and third returns to freshwater of coastal cutthroat trout in small coastal streams is low. Under the best survival rates, estimated recruitment potential from egg deposition to smolt ranged from 5 to 79 fish, smolt to first return ranged from 2 to 28 fish, and first return to second return ranged from 1 to 14 fish. Under the worst survival rates, estimated potential recruitment values were 3 to 44, <1 to 6, and <1, respectively.

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[2]Cooperators included the U.S. Fish and Wildlife Service, Washington Department of Game, and the University of Washington. The Unit has since been reassigned to the U.S. Geological Survey. The Washington Department of Game became a part of the Washington Department of Fish and Wildlife through reorganization.

INTRODUCTION

Coastal cutthroat trout, *Oncorhynchus clarki clarki*, inhabit lakes, streams, rivers, and estuaries along the Pacific coast from the Eel River in northern California, north along the coasts of Oregon, Washington, and British Columbia, to Prince William Sound in Alaska, rarely occurring farther inland than 150-160 km (Behnke 1992; Trotter 1989, 1997; Northcote 1997). Anadromous trout spend only a short time in saltwater that may be confined in the estuaries of their natal streams or inshore areas (Trotter 1989, 1997). However, some coastal cutthroat trout were captured with purse seines in the Columbia River plume, primarily between 10-50 km offshore (Pearcy 1997). Certain stocks have declined in the natural range of this subspecies from human land-use practices related to dams, culverts, water diversions, logging, road building, mining, livestock grazing, and urbanization, behavior interactions with other salmonids, and fishing mortality (Nehlsen et al. 1991; Gerstung 1997; Hooton 1997; Leider 1997; Reeves et al. 1997; Slaney et al. 1997; Williams and Nehlsen 1997; Connolly and Hall 1999; Johnson et al. 1999; Blakley et al. 2000).

The anadromous form of this subspecies typically exhibits high fidelity to spawn in small (1st and 2nd order), isolated headwater tributaries of coastal streams (Moring and Lantz 1975; Johnston and Mercer 1976[3]; Jones 1977[4]; Johnston 1982; Michael 1983; Michael 1989; Pauley et al. 1989; Trotter 1989; Nickelson et al. 1992[5]; Hooton 1997; Trotter 1997; Johnson et al. 1999). Four life-history forms (anadromous, adfluvial, fluvial, and resident) may be present sympatrically in the same coastal or Columbia River drainage (Hooton 1997; Blakley et al. 2000) but the behavioral relationships among the four forms are not clearly understood (Hooton 1997). Identification of individual coastal cutthroat trout stocks is more difficult than for other Pacific Northwest salmonids because this subspecies has not been studied in enough detail in most parts of its range to designate discrete stocks with certainty (Blakley et al. 2000). In Washington, forty stock complexes of coastal cutthroat trout were identified (Blakley et al. 2000). A stock complex was defined as "A group of stocks typically located within a single watershed or other relatively limited geographic area and believed to be closely related to one another." Hybrids between coastal cutthroat trout and steelhead have been reported where the two species spawn in the same stream reach (Campton and Utter 1985; Johnson et al. 1999; Young et al. 2001). However, steelhead typically spawn and rear downstream of cutthroat trout spawning areas that tends to reduce interactions between the two species (Hartman and Gill 1968). Hybrids between these two species can grow to maturity and reproduce but adults are not generally common since most

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[3]Johnston, J.M. and S.P. Mercer. 1976. Sea-run cutthroat in saltwater pens: Broodstock development and extended juvenile rearing (with a life history compendium). Washington State Game Department, Federal Aid Project AFS-57-1. 92 pp.

[4]Jones, D.E. 1977. Life history of sea-run cutthroat trout. Alaska Department of Fish and Game, Juneau. Completion report, Study Number AFS-42:78-105.

[5]Nickelson, T.E., J.W. Nicholas, A.M. McGie, R.B. Lindsay, and D.L. Bottom. 1992. Status of anadromous salmonids in Oregon coastal basins. Research and Development Section, Oregon Department of Fish and Wildlife, Corvallis. 81 pp.

hybrids are detected only in young (age 0 and 1) fish (Johnson et al. 1999). Reproductive isolation may occur among different life history forms or species through assortative mating (Hanson and Smith 1967; Sargent et al. 1986; Foote and Larkin 1988), or restricted gene flow (Williams et al. 1997; Johnson et al. 1999; Wenburg and Bentzen 2001).

Although various life history reviews have been published on coastal cutthroat trout (Giger 1972; Johnston and Mercer 1976[3]; Pauley et al. 1989; Trotter 1989, 1997; Johnson et al. 1999), there is a paucity of knowledge related to fecundity of this subspecies that is critical in estimating recruitment from natural reproduction (Myers and Barrowman 1996). This paper describes fecundity of coastal cutthroat trout from selected Oregon and Washington waters as related to fish size (fork length and weight) and estimates the recruitment potential of this species for three Oregon and two Washington streams.

MATERIALS AND METHODS

Fecundity

Fecundity of most female coastal cutthroat trout used in this study were believed to be anadromous fish because they were collected in the lower reaches of coastal rivers or lakes close to saltwater during migrations in Oregon and Washington between September and December. Anadromous trout had access to Barrett Lake, Washington via Tenmile Creek, a tributary to the Nooksack River. Six females from an adfluvial stock in Klickitat Lake, Oregon (Alsea River watershed) were included in the analysis of Oregon stocks to provide fecundity values for smaller fish. Since the spawning seasons of the four forms overlap (Blakley et al. 2000), it is possible that genetic exchange occurs between the different life-history forms so other trout from the different life-history forms may have been included in this study. In addition, trout produced at the Alsea River Hatchery were stocked in the Alsea and Siuslaw Rivers, Oregon and may have been included with the females collected in those rivers. The broodstock was developed from wild fish from the Alsea River, Oregon.

Coastal cutthroat trout were obtained from some anglers and various investigators who collected trout while conducting other studies. Fecundity data for Oregon fish were collected during the fall of 1968-69 and for Washington fish during the fall of 1969-1972 (most during the falls of 1969-70 and 1970-71). Females were measured to the nearest millimeter (mm) of fork length (FL) and weighed to the nearest gram (g). Ovaries were removed from the females, wrapped in cheesecloth, labelled, and preserved in 10% formalin. Fecundity was determined as the total number of maturing eggs contained in an mature female from direct counts of maturing eggs or ripe eggs that were easily recognized from immature eggs by their large size (2 to 5 mm in diameter) and yolk coloration (yellow to orange). Egg size and coloration was related to the date of capture. Differences in fecundity are generally correlated to length, weight, and age among individual fish (Bagenal 1978). Length is considered the most reliable parameter to estimate fecundity, followed by weight, and age is often the least useful predictor of fecundity due to high variability in annual growth among fish.

Variation in fecundity was comparable for females of a given size (fork length and weight) from various waters in Oregon and Washington in scatter diagrams so the data from each state were combined for analysis. Twelve ripe trout from the Beaver Creek Hatchery and 13 ripe trout from the Cowlitz Trout Hatchery, Washington, autopsied to obtain fecundity, were included in this study because no statistical difference (95% confidence level) was found between average fecundity of hatchery fish from the Beaver and Cowlitz River hatcheries (Wydoski and Tipping 2001) and other trout used in this study.

Regression equations were calculated between fork length and weight with fecundity for individual coastal cutthroat trout from Oregon and Washington using statistical procedures for regression analyses (e.g., Simpson et al. 1960; Snedecor and Cochran 1980). Regression equations were also calculated between weight and fecundity for Oregon and Washington trout. Although fecundity may vary in different years for different fish stocks, regression analyses averages any variation. The relationship of fecundity and fish size was analyzed and compared for trout collected in Oregon with those collected in Washington.

Predicted values of fecundity and 95% confidence intervals were calculated from regression equations for grouped intervals of fork length and weight (SPSS, Inc. 1997). Fecundity reported by total lengths (TL) of coastal cutthroat trout in other studies (Cramer 1940; Jones 1977[4]) were converted to fork lengths (FL) for comparisons with the results of this study using the regression equation FL = TL - 5.34 / 1.023 (n = 181; r = 0.995) from Wydoski and Tipping (2001).

Recruitment Potential

Recruitment potential for coastal cutthroat trout was determined for coastal cutthroat trout from three Oregon and two Washington intensively studied streams with the only available data on upstream movement of trout (Moring and Lantz 1975; Michael 1989). The average numbers and sizes of in-migrant trout were obtained from those studies. The number of females among in-migrant trout was obtained for each stream using sex ratios of the migrants. Estimates of fecundity were calculated for average lengths of in-migrant trout from regression equations in Figures 1 (Oregon) and 3 (Washington), then expanded to annual egg deposition from the annual average number of in-migrant adult females in three Oregon and two Washington streams.

The average number of in-migrant adult trout, sex ratios, and average size of in-migrant trout by years of ocean life in Deer Creek, Flynn Creek, and Needle Branch were obtained from a report summarizing a 15-year study (1959-1973) on the effects of logging in Alsea River, Oregon watershed (Moring and Lantz 1975). The number of adult in-migrant females was determined using sex ratios of 1.8 males: 1 female and 1.6 males: 1 female provided by Moring and Lantz (1975) for Deer and Flynn creeks, respectively. The average sex ratio of 1.7 males: 1 female for Deer and Flynn Creeks was applied to Needle Branch since the sex ratio was not provided. The average sizes of in-migrant trout by years of ocean life were obtained from Table 25 of Moring and Lantz (1975). The average percentage of trout returning after 1, 2, and 3 ocean years was 71.4%, 25.4%,

and 3.3% for Flynn Creek, and 73.9%, 24.6%, and 1.4% for Deer Creek (Table 22, Moring and Lantz 1975). The average value for Deer and Flynn creeks of 72.7%, 25.0%, and 2.4% for 1, 2, and 3 ocean years, respectively, was applied to Needle Branch. The average size of fish with 1, 2, or 3 years of ocean history was 337, 401, and 424 mm fork length, respectively (Moring and Lantz 1975).

The average number and average size of in-migrant adult trout in Salmon Creek, Washington (4 years; 1977-1981) and in Snow Creek, Washington (3 years; 1977-1980) were obtained from an 8-year study by Michael (1989). Since sex ratios were not provided by Michael (1989), the number of adult females was estimated using the average sex ratio of 1.7 males: 1 female from Moring and Lantz (1975). The proportion of trout with 1, 2, and 3 ocean years for Salmon and Snow creeks was 69.4%, 27.4%, and 3.2%, respectively (Table 3, Michael 1989). The average fork lengths of first, second, and third return fish from the ocean in Salmon and Snow creeks were 303, 402, and 460 mm, respectively (Table 3, Michael 1989).

Survival rates of 0.53-0.95% from egg deposition to out-migrant smolts, 10-50% from out-migrant smolts to the first freshwater return, and 5-50% from the first freshwater return to the second freshwater return were applied to the three Oregon streams from the salmonid literature review by Johnston (1979)[6]. The same survival rates were used from egg deposition to out-migrant smolts and first return to second freshwater return for Washington streams except for smolt to first freshwater return (13.9-21.7% for Salmon Creek and 1.8-4.3% for Snow Creek) reported by Michael (1989: 192).

Numbers of smolt, first freshwater return, and second freshwater return trout were calculated from those survival rates to estimate the recruitment potential for in-migrant adult trout in the three Oregon and two Washington streams. The numbers of trout having three years of ocean life were low so they were not included in the analyses.

RESULTS

Differences in Fecundity Between Washington and Oregon Trout

The fecundity of coastal cutthroat trout from Washington fish (n = 88) was significantly higher than for Oregon fish (n = 90) for both fork length and weight. The slopes ($F = 53.73$, $p < 0.01$) and y-intercepts ($F = 220.62$, $p < 0.01$) of the regression equations for fecundity with fork length were found to be significantly different through covariance analysis. The slopes ($F = 3.96$, $p < 0.05$) and y-intercepts ($F = 46.54$, $p < 0.01$) for fecundity with weight were also significantly different between Oregon (n = 84) and Washington (n = 76) females. Washington trout (Figure 3) contained an average of 131 more eggs in females 250 mm FL to 227 more eggs in females 500 mm FL than Oregon trout (Figure 1). Washington trout (Figure 4) also contained an average of 372 more eggs for 200 g fish and 103 more eggs for 1,200 g fish than Oregon trout (Figure 2).

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[6]Johnston, J.M. 1979. Sea-run cutthroat: Stillaguamish River creel census (1978). Washington State Game Department, Fish Management Division, Olympia. 15 pp.

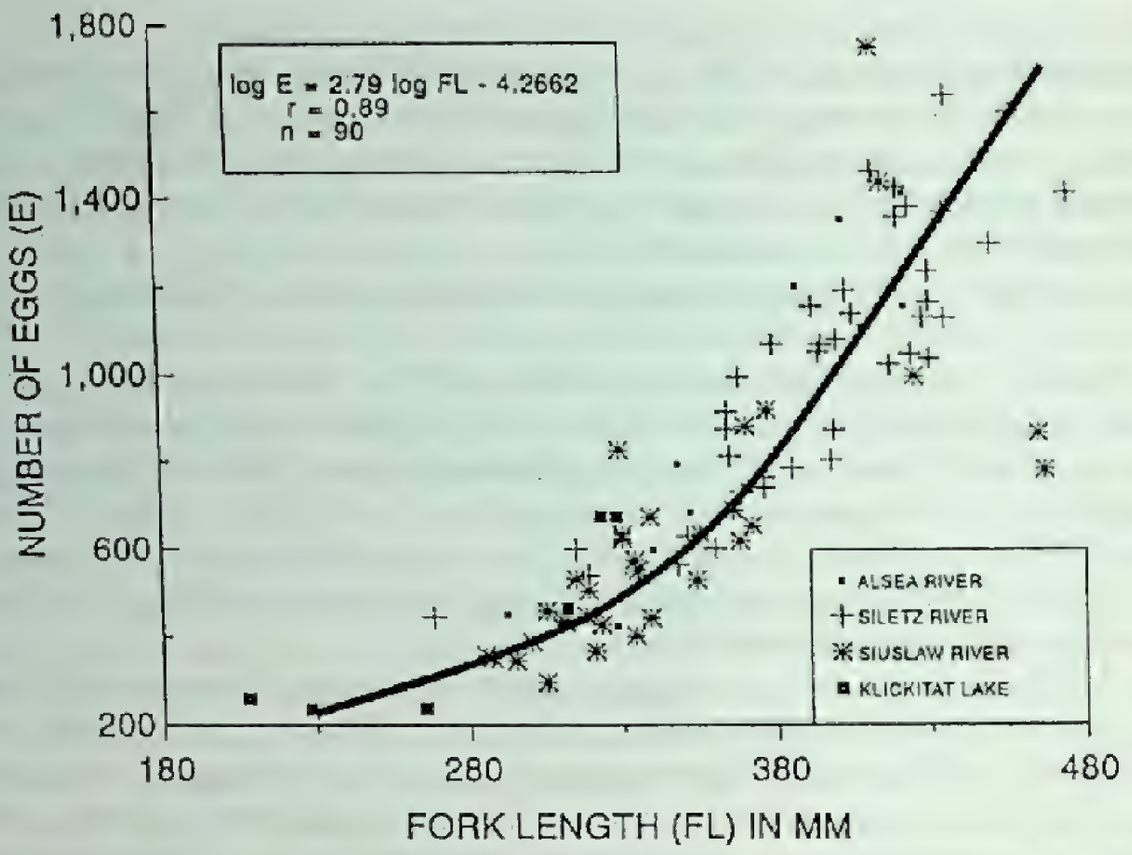

Figure 1. Relation of fecundity to fork length for coastal cutthroat trout from Oregon.

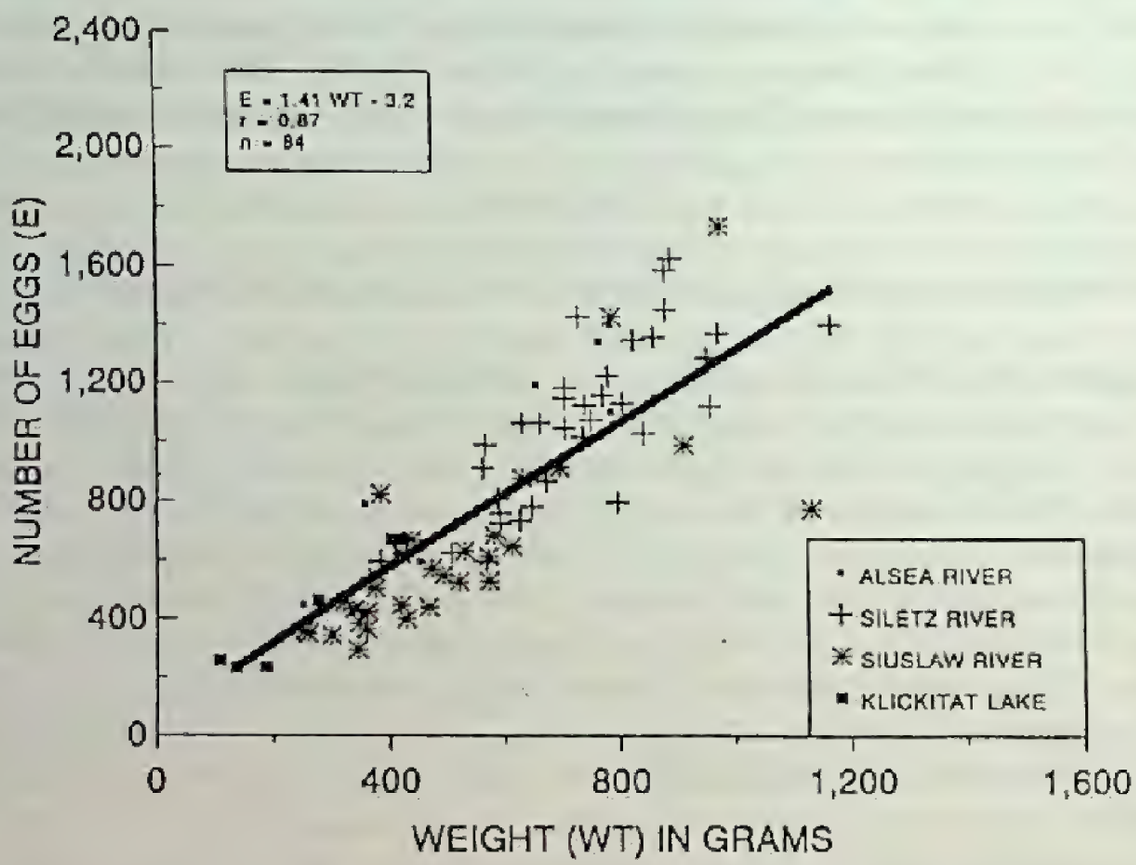

Figure 2. Relation of fecundity to weight for coastal cutthroat trout from Oregon.

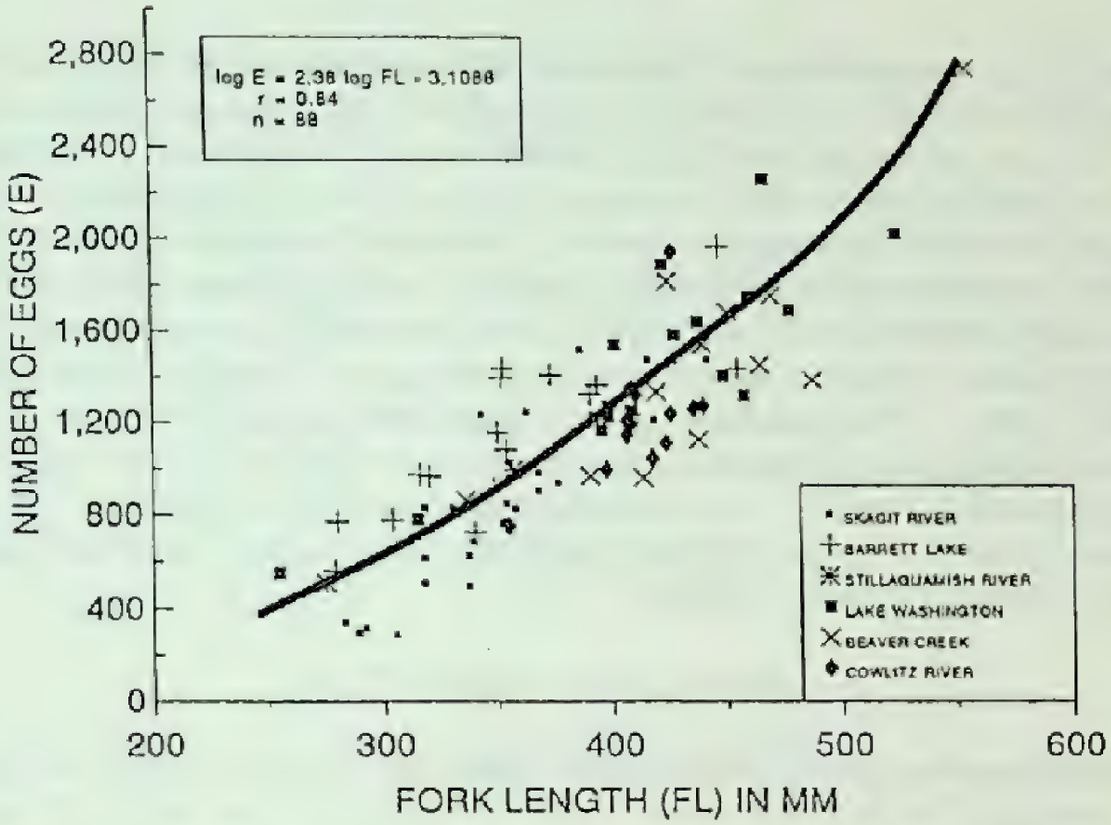

Figure 3. Relation of fecundity to fork length for coastal cutthroat trout from Washington.

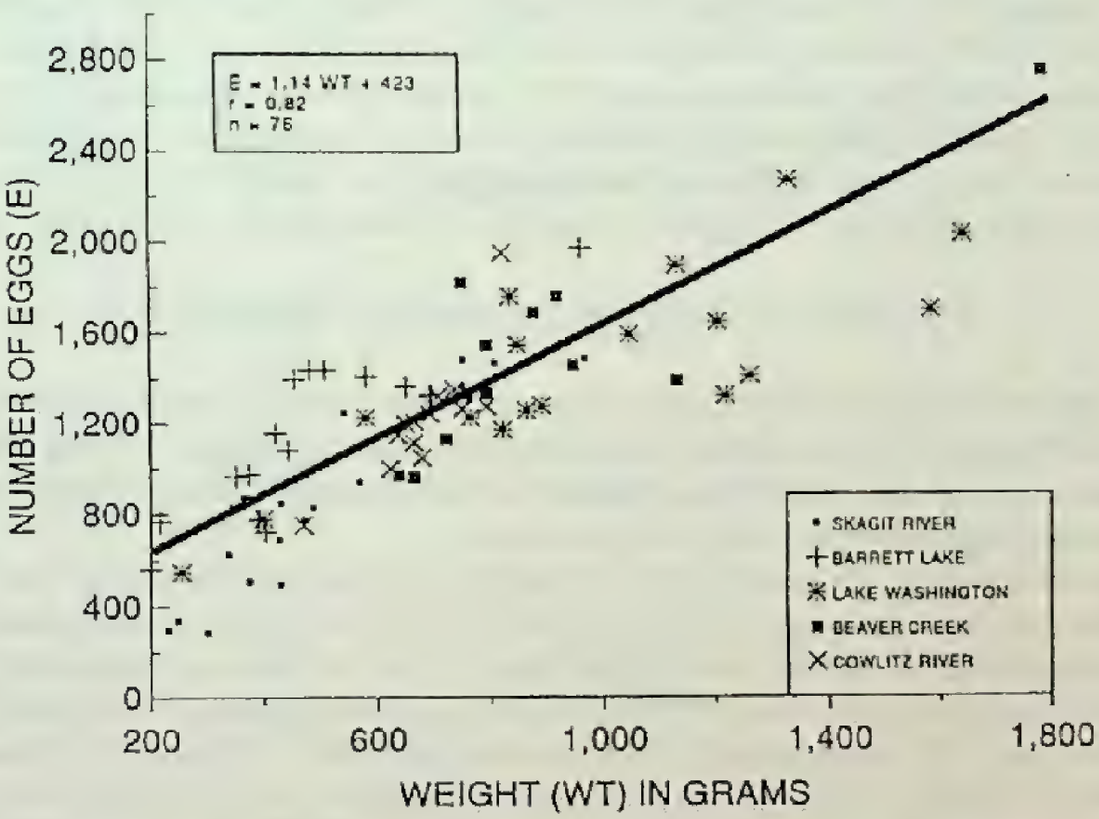

Figure 4. Relation of fecundity to weight for coastal cutthroat trout from Washington.

Fecundity of Oregon Trout

The average number of eggs produced by coastal cutthroat trout from Oregon was 802 eggs (SE = 40.09) for fish that were 364 mm (SE = 6.0; n = 90) in average FL and 576 g (SE = 26.3; n = 84) in average weight. Fecundity of individual females from Oregon waters increased in concert with increasing length as well as with increasing weight, although there was a wide range in fecundity for females of similar sizes (Figures 1 and 2). The estimated fecundity of individual trout increased from an average of 160 eggs for 208 mm FL females to 1,591 eggs for 475 mm FL females (Figure 1). The estimated fecundity for individual trout increased from an average of 149 eggs for 108 g females to 1,647 eggs for 1,170 g females (Figure 2). Predicted average fecundity with 95% confidence intervals for grouped data are summarized by length intervals (Table 1) and by weight intervals (Table 2). The predicted average fecundity increased from 138 eggs for 218 mm fish to 1,276 eggs for 466 mm fish (Table 1) and from 421 eggs for 273 g fish to 1,556 eggs for 1,152 g fish (Table 2).

Fecundity of Washington Trout

The average number of eggs produced by coastal cutthroat trout from Washington was 1,108 eggs (SE = 56.55) for fish that were 372 mm (SE = 7.3; n = 88) in average FL and 661 g (SE = 49.5; n = 76) in average weight. Although there was a wide range in fecundity for females of similar sizes, average fecundity of individual Washington females increased in concert with increasing length (Figure 2) as well as with increasing weight (Figure 4). The average fecundity for individual Washington trout increased from 416 eggs for a 255 mm fish to 2,318 eggs for a 525 mm fish (Figure 3) and from 651 eggs for a 200 g fish to 2,297 eggs for a 1,644 g fish (Figure 4). Predicted average fecundity with 95% confidence intervals for grouped data are summarized by length intervals (Table 1) and by weight intervals (Table 2). The predicted average fecundity increased from 515 eggs for fish with a mean length of 265 mm to 1,596 eggs for 466 mm fish (Table 1) and from 716 eggs for 234 g fish to 2,240 eggs for 1,715 g fish (Table 2).

Recruitment Potential of Coastal Cutthroat Trout

Egg deposition and recruitment potential of anadromous coastal cutthroat trout was estimated for three Oregon and two Washington streams. These estimates demonstrate that the recruitment potential of adult coastal cutthroat trout returning from saltwater into small coastal streams is low.

In an Oregon study on the effects of logging, Flynn Creek (2,493 m^2 site) was a control stream, Deer Creek (4,183 m^2 site) had three 25-ha patch cuts with a buffer strip left along the main channel, and Needle Branch (1,063 m^2 site) was clearcut without a buffer strip (Moring and Lantz 1975). The average annual egg deposition for Deer Creek, Flynn Creek, and Needle Branch was estimated to be 5,836, 4,679, and 545, respectively (Table 3). Based on the highest survival rates from Johnston (1979)[6], an average of 5-55 smolts would be produced, 3-28 smolts surviving to the first freshwater return, and

Table 1. Predicted fecundity (total number of eggs) for coastal cutthroat trout from grouped fork length data for Oregon and Washington.

	Oregon					Washington				
Fork length interval (mm)	Fork length Mean (mm)	Fork length Standard error	Number of fish	Number of eggs[a] Predicted average	95% CI[c]	Fork length Mean (mm)	Fork length Standard error	Number of fish	Number of eggs[b] Predicted average	95% CI[c]
201 - 250	218.0	10.00	2	121	0 - 299	-	-	-	-	-
251 - 275	266.5	1.50	2	350	219 - 481	265.0	10.00	2	452	324 - 581
276 - 300	291.8	2.71	5	466	355 - 577	284.6	2.54	5	564	452 - 677
301 - 325	314.0	1.91	14	569	471 - 666	313.4	2.34	10	721	629 - 814
326 - 350	335.1	1.82	15	667	577 - 756	337.5	1.80	8	861	783 - 940
351 - 375	362.4	1.84	14	793	703 - 882	358.4	1.89	15	974	902 - 1,046
376 - 400	387.1	2.72	12	909	810 - 1,008	391.9	2.01	14	1,165	1,093 - 1,236
401 - 425	414.5	2.02	15	1,040	922 - 1,157	412.9	1.80	15	1,282	1,203 - 1,362
426 - 450	434.7	3.28	7	1,133	999 - 1,268	438.9	2.41	9	1,428	1,334 - 1,523
451 - 475	466.0	3.70	4	1,278	1,114 - 1,442	460.1	2.35	8	1,546	1,437 - 1,655
476 - 500	-	-	-	-	-	483.0	5.00	2	1,675	1,548 - 1,802

[a]Calculated from the regression equation TE = 4.67 FL - 896; n = 10 (90 fish); r = 0.96; F = 82.99; p = < 0.0001.
[b]Calculated from the regression equation TE = 5.61 FL - 1,035; n = 10 (88 fish); r = 0.98; F = 173.31; p = <0.0001.
[c]CI = confidence interval

Table 2. Predicted fecundity (total number of eggs) for coastal cutthroat trout from grouped weight data for Oregon and Washington.

	Oregon					Washington				
Weight interval (g)	Mean (g)	Weight Standard error	Number of fish	Number of eggs[a] Predicted average	95% CI[c]	Mean (g)	Weight Standard error	Number of fish	Number of eggs[b] Predicted mean	95% CI[c]
201 - 300	272.5	10.66	4	421	256 - 586	233.5	12.25	4	716	474 - 958
301 - 400	351.2	5.18	13	530	388 - 672	360.5	10.17	9	845	635 - 1,057
401 - 500	442.4	7.99	14	640	518 - 761	442.7	8.66	11	931	739 - 1,124
501 - 600	565.2	8.25	14	798	680 - 899	556.8	13.70	7	1,049	878 - 1,219
601 - 700	651.6	8.77	11	911	817 - 1,005	659.3	6.62	12	1,154	999 - 1,308
701 - 800	757.7	8.81	12	1,048	949 - 1,181	757.1	7.90	12	1,255	1,110 - 1,399
801 - 900	860.0	10.48	8	1,179	1,065 - 1,294	846.9	10.66	8	1,347	1,207 - 1,487
901 - 1,000	956.8	11.65	6	1,304	1,168 - 1,441	951.7	15.43	3	1,455	1,314 - 1,597
1,001 - 1,200	1,152.0	18.00	2	1,556	1,366 - 1,747	1,105.7	28.33	3	1,613	1,456 - 1,770
1,201 - 1,400	-	-	-	-	-	1,254.5	28.54	4	1,767	1,584 - 1,950
1,401 - 1,600	-	-	-	-	-	1,587.0	71.00	2	2,109	1,848 - 2,370
1,601 - 1,800	-	-	-	-	-	1,715.0	-	1	2,240	1,945 - 2,537

[a]Calculated from the regression equation TE = 1.29 WT ± 68.9; n = 9 (84 fish); r = 0.96; F = 79.43; p < 0.0001.
[b]Calculated from the regression equation TE = 1.03 WT ± 475.1; n = 12 (76 fish); r = 0.0.92; F = 55.1; p <0.0001.
[c]CI = confidence interval

Table 3. Estimated annual egg deposition and potential recruitment of adult anadromous coastal cutthroat trout from Oregon and Washington streams.

Stream	Average annual number of in-migrant adult trout	Average number of adult females	Average FL (mm)	Average fecundity per female[a]	Calculated annual egg deposition	Number of fish from egg deposition to adults based on survival rates[b]		
						Smolt 0.53-0.95%	1st Return 10-50%	2nd Return 5-50%
Deer Creek OR	22.7[c]	8.1[d]						
1 yr. ocean		6.0	337[e]	618	3,708			
2 yrs. ocean		2.0	401[e]	1,000	2,000			
3 yrs. ocean		0.11	424[e]	1,160	128			
					5,836	31-55	3-28	<1-14
Flynn Creek OR	16.7[c]	6.45[d]						
1 yr. ocean		4.7	337[e]	618	2,905			
2 yrs. ocean		1.6	401[e]	1,000	1,600			
3 yrs. ocean		0.15	424[e]	1,160	174			
					4,679	25-45	2-23	<1-12
Needle Branch OR	2.0[c]	0.74[d]						
1 yr. ocean		0.54	337[e]	618	334			
2 yrs. ocean		0.19	401[e]	1,000	190			
3 yrs. ocean		0.018	424[e]	1,160	21			
					545	3-5	<1-3	<1-2

Stream	Average annual number of in-migrant adult trout	Average number of adult females	Average FL (mm)	Average fecundity per female[a]	Calculated annual egg deposition	Number of fish from egg deposition to adults based on survival rates[b]		
						Smolt 0.53-0.95%	1st Return 10-50%	2nd Return 5-50%
Salmon Creek WA[f]	27.5[g]	10.2[h]						
1 yr. ocean		7.1	303[i]	627	4,452			
2 yrs. ocean		2.8	402[i]	1,200	3,360			
3 yrs. ocean		0.33	460[i]	1,670	551			
					8,363	44-79	6-17	<1-9
Snow Creek WA[j]	14.3[g]	5.3[h]						
1 yr. ocean		3.7	303[i]	627	2,330			
2 yrs. ocean		1.5	402[i]	1,200	1,800			
3 yrs. ocean		0.17	460[i]	1,670	284			
					4,404	23-42	<1-2	<1-1

[a]Fecundity was determined from the average fork length of trout from the regression equations in Figure 1 for Oregon and in Figure 3 for Washington.
[b]Survival rates for Oregon fish are from Johnston (1979)[6]: 0.53-0.95% from egg deposition to smolt; 10-50% from smolt to first return to freshwater, and 5-50% from first return to second return to freshwater.
[c]Average number of in-migrant adult trout for Oregon from Appendix 8, Moring and Lantz (1975).
[d]The number of in-migrant adult female trout was based on sex ratios in upstream traps reported by Moring and Lantz (1975) for Deer Creek (1.8 males:1 female) and Flynn Creek (1.6 males:1 female). The average sex ratio of 1.7 males:1 female) for Deer and Flynn creeks was applied to Needle Branch since the sex ratio was not reported for that stream by Moring and Lantz (1975).
[e]Average size of in-migrant trout in Oregon from Table 25, Moring and Lantz (1975).
[f]Survival rates for Salmon Creek fish were also from Johnston (1979)[6] except for 13.9-21.7% survival from smolt to first return reported by Michael (1989).
[g]Average number of in-migrant adult trout for Washington from Table 1 of Michael (1989).
[h]Average number of in-migrant adult female trout was estimated from a sex ratio of 1.7 males: 1 female that was the average value for Oregon fish.
[i]Average size of in-migrant trout in Washington from Table 3 of Michael (1989).
[j]Survival rates for Snow Creek fish were also from Johnston (1979)[6] except for 1.8-4.3% survival from smolt to first return reported by Michael (1989).

2-14 trout surviving from the first to the second freshwater return in the three Oregon streams (Table 3). At the lowest survival rates, an average of 3-31 smolts would be produced, <1-3 smolts surviving to the first freshwater return, and <1 trout surviving from the first to the second freshwater return in the three streams (Table 3). The lowest trout production, even when adjusted for size of the three Oregon study sites, was in Needle Branch where the watershed was clearcut. Not only did the numbers of coastal cutthroat trout generally decrease in Oregon stream basins that were extensively clearcut such as Needle Branch, but trout numbers remained low for up to 60 years after being clearcut unless the stream habitats were restored by natural occurrences or human restoration efforts (Connolly and Hall 1999).

In Washington, the recruitment potential of coastal cutthroat trout was estimated for Salmon and Snow creeks (Michael 1989). Both streams drain into the Strait of Juan de Fuca on the north side of the Olympic Peninsula. The average annual egg deposition for Salmon Creek and Snow Creek was estimated to be 8,363 and 4,404, respectively (Table 3). At the highest survival rates, an average of 42-79 smolts are produced, 2-17 smolts surviving to the first freshwater return, and 1-9 trout surviving from the first to the second freshwater return in these two streams (Table 3). At the lowest survival rates, 23-44 smolts are produced, <1-6 smolts surviving to the first freshwater return in each stream, and <1 trout surviving from the first to the second freshwater return (Table 3).

DISCUSSION

Differences in Fecundity Between Oregon and Washington Trout

Some anadromous salmonids along the Pacific coast exhibit relatively small differences in fecundity between stocks from streams at the same latitude but tended to increase significantly from southern to northern latitudes (Drucker 1972; Fleming and Gross 1990). For example, the mean fecundity of coho salmon *Oncorhynchus kisutch* stocks increased from about 2,000 eggs for a stock at 44-45 ° N latitude to over 4,000 eggs for stocks at 60° N latitude (Drucker 1972). While fecundity increased significantly from southern to northern latitudes for 17 stocks of coho salmon on the Pacific coast, egg size decreased significantly with latitude in these stocks (Fleming and Gross 1990). Egg number and egg size have been described as opposing attributes in which selection for one attribute results in a trade-off for the other (Svardson 1949).

Coastal cutthroat trout from Oregon were collected from waters at about 44-45° N latitude while samples from Washington fish were obtained from cutthroat collected from waters at about 48-49° N latitude. A small, but statistically significant difference was found in the regression equations for fecundity of coastal cutthroat trout from Oregon and Washington. However, other factors may have influenced the difference in fecundity between Oregon and Washington including samples from different waters, sample sizes, and differences in years that the trout were collected. The relation between fecundity and latitude did not hold for coastal cutthroat trout from Petersburg Creek in southeastern Alaska. Fecundity was lower for smaller females but higher for

larger fish from Alaska (~ 54° N latitude) when compared with values for Oregon and Washington trout (Jones 1977[4]).

Variation and Similarities in Fecundity for Coastal Cutthroat Trout from Different Locations

Coastal cutthroat trout from different Oregon (Figures 1 and 2) or Washington waters (Figures 3 and 4) exhibited a wide range in fecundity for females of similar lengths or weights. However, the range of fecundity for this subspecies from different waters was found to be remarkably similar for fish with similar lengths or weights in both Oregon (Figures 1 and 3) and Washington (Figures 2 and 4). This study suggests that size of coastal cutthroat trout is most closely related to fecundity and that the variation in fecundity from comparable sizes of fish from different waters (i.e., stocks) is similar.

Comparison of Fecundity in Other Studies

Few studies have been reported on the fecundity of wild coastal cutthroat trout. Cramer (1940) reported fecundity of two anadromous and 5 resident coastal cutthroat trout. One anadromous female (380 mm FL) contained 1,098 eggs and another fish (371 mm FL) contained 1,242 eggs. Resident fish produced fewer eggs: 415 eggs in a 247-mm FL fish, 380 in a 273-mm fish, and 615 eggs in a 311 mm fish. The lengths of the other two resident fish were not stated, but one fish contained 467 eggs and the other fish contained 521 eggs. Jones (1977)[4] reported an average fecundity of 873 eggs from 22 female anadromous coastal cutthroat trout from Petersburg Creek, Alaska. The range in fecundity for Petersburg Creek fish was 460 eggs for a fish 340 mm FL to 2,342 eggs for a 422 mm fish. Jones provided the fecundity by length for four other female trout in earlier (1975 and 1976) reports: 486 eggs for a 340 mm fish, 500 eggs for a 370 mm fish, 1,100 eggs for a 440 mm fish, and 2,286 eggs for a 460 mm fish.

Fecundity reported by Cramer (1940) for two anadromous coastal cutthroat trout in Oregon was slightly higher for fish of comparable lengths than fecundity determined for Oregon fish in this study (Figure 1). The fecundity reported by Jones (1977)[4] is lower for smaller females but higher for larger fish when compared with values for Oregon and Washington trout. Estimated fecundity of Oregon fish would be 629 eggs for a 340-mm fish, 1,152 eggs for a 422-mm fish, 1,285 eggs for a 440-mm fish, and 1,455 eggs for a 460-mm fish (Figure 1) while estimated fecundity of Washington fish would be 827 eggs for a 340-mm fish, 1,386 eggs for a 422-mm fish, 1,523 eggs for a 440-mm fish, and 1,693 eggs for a 460-mm fish (Figure 3).

No significant difference in fecundity (95% confidence level) was found for data grouped by 25 mm of length and 100 g of weight for coastal cutthroat trout at the Beaver Creek and Cowitz River, Washington hatcheries (Wydoski and Tipping 2001) when compared to the fish from Washington in this study. However, fecundity in trout at the Alsea River Hatchery, Oregon (Wydoski and Tipping 2001) was significantly higher (95% confidence level) when compared to the fish from Oregon in this study. The higher fecundity in fish from the Alsea River Hatchery is attributed to faster growth of

broodstock maintained at the hatchery. An increase in the quantity and quality of food was demonstrated to increase growth and fecundity was related to growth rate in other salmonids (Scott 1962; Bagnal 1969a). The broodstock at the Alsea River Hatchery was developed and maintained using wild coastal cutthroat trout from the Alsea River.

Recruitment Potential of Coastal Cutthroat Trout

Recruitment potential for three Oregon and two Washington streams demonstrated that estimated survival from egg deposition to smolt, smolt to first return in freshwater, and second return in freshwater of coastal cutthroat trout in small coastal streams is low. These results are supported by densities between 0.02 and 0.24 fish per m^2 for young-of-the-year coastal cutthroat trout in Washington streams and between 0.00 and 0.31 fish per m^2 for age 1+ trout in Oregon and Washington streams (Moring and Lantz 1975; Phillips et al. 1981[7]; Mercer 1982[8]; and Peoples et al. 1988[9]). Recruitment potential may vary considerably due to variation in survival rates at different life stages. In addition, the annual out-migration of smolts and returns of adults varies considerably in all parts of the coastal cutthroat trout range (e.g., Moring and Lantz 1975; Johnston and Mercer 1976[3]; Jones 1977[4]; Michael 1983; Michael 1989; Johnson et al. 1999; Blakley et al. 2000). This variation in out-migration can be explained by the annual variations throughout the range of coastal cutthroat trout in number of repeat out-migrant adults and growth rates of individual smolts that influences their ages when they become large enough to migrate.

Potential Recruitment by Size and Age of Coastal Cutthroat Trout

Larger salmonid females produce more and larger eggs (Rounsefell 1957; Sargent et al. 1987), select and defend better habitats for redds (van den Berghe 1984[10]) that develop into larger alevins and fry (Beacham and Murray 1990), giving their offspring a competitive advantage in survival over offspring from smaller fish (Bagenal 1969b). Since larger coastal cutthroat trout females spawn earlier than smaller females (Giger 1972; Tipping 1981[11]) and produce larger eggs (Wydoski and Tipping 2001) that

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[7]Phillips, C., R. Cooper, and T. Quinn. 1981. Skagit River salmonid studies, 1977-1981, Washington Department of Game and U.S. Fish and Wildlife Service, Olympia. Final Report 82-1. 104 pp.

[8]Mercer, S.P. 1982. Sea-run cutthroat: Broodstock development and evaluation techniques. Washington State Game Department, Olympia. Federal Aid Proj. F-68-R. 55 pp.

[9]Peoples, M., M. Cenci, and J. Hunter 1988. Sea-run cutthroat studies in south Puget Sound tributaries, July 1986 to June 1987. Washington Department of Wildlife, Olympia. Fish Management Division Report 88-12. 39 pp.

[10]van den Berghe, E.P. 1984. Natural selection and reproductive success in female coho salmon (*Oncorhynchus kisutch*): a study of female competition. M.S. Thesis, Simon Fraser University, Burnaby, BC, Canada.

[11]Tipping, J.M. 1981. Cowlitz sea-run cutthroat study. Washington State Game Department, Olympia. Fish Management Division Report 81-12. 26 pp.

probably hatch as larger alevins, their progeny can be expected to have a higher survival because they will likely migrate as larger smolts. Smolt-to-adult survival increased with increasing smolt length in numerous salmonids including coastal cutthroat trout (Tipping 1986; Tipping and Blankenship 1993); pink salmon, *Oncorhynchus gorbuscha* (Taylor 1980); coho salmon (Hager and Noble 1976; Bilton et al. 1982), steelhead trout, *Oncorhynchus mykiss* (Ward and Slaney 1988; Ward et al. 1989; Tipping 1997), sockeye salmon, *Oncorhynchus nerka* (Forester 1954; Bradford and Peterman 1987; MacDonald et al. 1987; West and Larkin 1987; Henderson and Cass 1991), and chinook salmon, *Oncorhynchus tshawytscha* (Fowler 1972; Bilton 1984; Martin and Wertheimer 1989). Although most smolt-to-adult survival has been reported for hatchery-reared salmonids, higher survival would also be expected for larger wild smolts. Stream-dwelling salmonids exhibit territoriality and aggressive behavior in which larger fry select and maintain prime habitats for feeding and escape cover (Chapman 1962; Mason and Chapman 1965; Chapman and Bjornn 1969; Mason 1976; Bachman 1984; Mesa 1991). Smaller salmonid fry are displaced by the aggressive behavior of larger fry that undoubtedly results in higher mortality of larger fry (Chapman 1962; Mason and Chapman 1965). While habitat quantity and quality influences the carrying capacity of streams for fish, earlier emergence, larger size of alevins, and higher expected survival of progeny from larger females may fill the carrying capacity in streams with anadromous salmonids such as coastal cutthroat trout. However, recruitment from smaller females is also important because more small fish return after the first ocean year than after the second ocean year. Likewise, more fish return after the second ocean year than after the third ocean year.

Relation of Fecundity to Management

Fecundities of coastal cutthroat trout from Oregon and Washington are adequate to provide higher recruitment but is presently limited by high mortality from egg deposition to the smolt stage and among adult trout from various factors including reduced habitat quantity and quality, increased adverse interactions with other salmonids, and angler harvest. Recruitment potential for trout from three Oregon and two Washington streams demonstrated that reproductive ability of coastal cutthroat trout is low. Anadromous trout, that spawn in small isolated headwaters of coastal streams, have been adversely impacted by habitat alterations (Hall and Lantz 1969; Bisson and Sedell 1984; Hall et al. 1987; Reeves et al. 1997; Connolly and Hall 1999), behavioral interactions with other salmonids (Andrusak and Northcote 1971; Giger 1972; Glova 1984; Glova and Mason 1976; Hartman and Gill 1968; Johnson et al. 1999; Nilsson and Northcote 1981), and angler pressure (Giger 1972; Gresswell and Harding 1997; Jones 1977; Leider 1997; Schmidt 1997), from headwater tributaries downstream through larger coastal streams. Adverse impacts from these factors even extend into estuaries and inshore marine areas. The low reproductive ability of coastal cutthroat trout is particularly important when considering that trout populations do not rebound quickly after adverse habitat alterations. For example, coastal cutthroat trout production generally decreased in basins that were clear-cut 20-60 years ago and may remain low

for the next 50 or more years unless the habitat is restored by favorable natural disturbances or habitat restoration (Connolly and Hall 1999). In addition, coastal cutthroat trout are probably more vulnerable to bird and marine mammal predation than other salmonids because they generally occupy water less than 3 m deep in estuaries or along protected beaches such as those found in Puget Sound (Johnston 1982). Effective management of coastal cutthroat trout stocks or stock complexes can be achieved through sound conservation measures and management actions that will increase survival at various life stages. Natural recruitment of coastal cutthroat trout can be increased by increasing habitat quantity and improving habitat quality for early life stages by regulating human habitat alterations, avoiding adverse interactions with other salmonids through prudent stocking practices of hatchery-reared fish, and protecting spawners through angling regulations.

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MARK LOSS RATE IN HATCHERY-REARED STRIPED BASS, *MORONE SAXATILIS*, IN THE SACRAMENTO-SAN JOAQUIN ESTUARY, CALIFORNIA

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Striped bass, *Morone saxatilis*, stocking in the Sacramento-San Joaquin Estuary was initiated in 1981 due to their declining abundance. Stocking success is determined by the survival and contribution of hatchery-reared striped bass to the sport fishery. Estimates of survival and contribution of these fish are based on mark-recapture methodology. Hatchery-reared, juvenile striped bass were marked with freeze brands (FB), fin clips (FC), and coded-wire microtags (CWT). In 1986 and 1989, striped bass were triple-marked to determine loss rates of all three marks. Results from the 1986 year-class indicated that CWT loss rate changed in 1990 when more efficient CWT detectors were used. The estimated CWT loss before 1990 was 16.7%; starting in 1990, estimated loss was 12.5%. Estimated FB/FC loss rate of the 1986 year-class increased at an annual rate of 11%, reaching 48% at age 7. Recoveries of marked fish from the 1989 year-class were too low to reliably estimate loss rates. These estimates were higher than what other studies have reported, but support previous findings that coded-wire tagging is an effective technique for marking hatchery-reared, juvenile striped bass.

INTRODUCTION

Striped bass, *Morone saxatilis*, were introduced into the Sacramento-San Joaquin Estuary (Estuary) from the East Coast in 1879 (Skinner 1962). Within 10 years, striped bass supported a major fishery in California. A commercial fishery existed until it was banned by legislation in 1935. The abundance of striped bass in the Estuary declined substantially between the mid-1960s and early 1980s (Stevens et al. 1985), and this decline continued into the early 1990s (Kohlhorst 1999). In 1981, the depressed state of the striped bass sport fishery prompted anglers to support legislation that required the purchase of a US$3.50 license stamp for striped bass fishing; revenue from the stamp has been used to restore the striped bass population. One facet of the restoration program was the stocking of hatchery-reared striped bass into the Estuary.

The stocking program began with fish from the 1980 year-class, which were stocked as yearlings in 1981. Studies to evaluate the effectiveness of the stocking program started with the 1981 year-class. The evaluation program used mark-recapture methodology to (1) determine the contribution of hatchery-reared fish to the legal-sized (45.7 cm total length, [TL]) striped bass population in the Estuary; (2) estimate survival of hatchery-reared striped bass from the time they are stocked until they reach legal size,

generally at age 3; and (3) determine the most effective management strategy for the striped bass hatchery program. From 1981 to 2001, more than 13 million hatchery-reared striped bass were released; 6 million of these were marked with freeze brands and fin clips (FB/FC), coded-wire microtags (CWTs), or both.

Estimates of survival and contribution of hatchery-reared striped bass to the sport fishery are dependent on unbiased recovery of marked fish; therefore, analyses must account for lost marks. Previous studies report that CWT loss from striped bass occurs within the first 2 weeks after tag implantation (Dunning et al. 1990, Wooley et al. 1990, Wallin and Van Den Avyle 1994). Various tag loss rates have been reported, e.g., 4% after 12 days (Wooley et al. 1990), 6–14% after 10 weeks (Wallin and Van Den Avyle 1994), 0% after 180 days (Klar and Parker 1986), and 4% after 29 months (N.C. Parker, USFWS, personal communication in Fletcher et al. 1987). These studies indicate CWT shedding rates stabilize over time.

Fin clips and freeze brands are regarded as appropriate marking techniques for short-term studies (McNeil and Crossman 1979, Boxrucker 1982), but may not be useful for long-term studies because, over time, fins regenerate and freeze brands fade. Black crappies, *Pomoxis nigromaculatus*, can be recognized as fin-clipped and freeze-branded for up to 31 weeks, though some fin regeneration occurred (Conover and Sheenan 1999). Elrod and Schneider (1986) found 20% of fin-clipped lake trout, *Salvelinus namaycush*, had more than half the fin present at age 1, but the fin had almost fully regenerated between ages 2 and 4. Fin-clipping largemouth bass, *Micropterus salmoides*, appeared to be effective for up to a year (Boxrucker 1982). Over a 6-month period, 91% of brook trout, *Salvelinus fontinalis*, could be distinguished as fin-clipped (Mears 1976). Freeze brands are recognizable for 4 months for bluegills, *Lepomis macrochirus*, (Coutant 1972); up to a year for salmonids (Knight 1990, Mighell 1969); and 40 months for walleyes, *Stizostedion vitreum* (Lajeon and Bergerhouse 1991).

We are aware of no study that documented the loss rate of CWTs, freeze-brands, or fin-clips over several years in the wild. We are reporting the estimated loss (including non-detection) rate of CWTs, fin clips, and freeze brands from hatchery-reared striped bass released at age 1 and recaptured up to age 7 in the Sacramento-San Joaquin Estuary. Loss rate information will indicate the most effective marking technique for hatchery-reared, juvenile striped bass and allow recoveries of marked fish to be corrected for loss and non-detection prior to data analysis.

METHODS

Marking

An ideal mass-marking method for striped bass uses easily applied marks that are detectable for many years, has no effect on growth or survival, and provides sufficient information on recaptures to meet the objectives of the marking program. The original technique selected in 1981 for this program was a combination FB/FC, which involved excising a single pelvic fin and branding each fish with a metal rod cooled with liquid nitrogen (Coutant 1972). Brands were placed posterior to the head, anterior to the dorsal

fin, and dorsal to the lateral line.

The FB/FC technique was replaced in 1984 with the present CWT system. This method involves the insertion of a stainless-steel wire microtag (1 x 0.25 mm) using a tag injector (Northwest Marine Technology[1], Shaw Island, Washington, Model Mark IV). Fingerlings are anesthetized with tricaine methane sulfonate (MS-222), and CWTs are inserted using a fully extended needle into the left adductor mandibularis (cheek musculature) in a dorso-ventral orientation 3–5 mm posterior to the eye (Fletcher et al. 1987). All tagged fish are passed through a quality control device to ensure tags are implanted. Those without CWTs are retagged.

In 1986 and 1989, striped bass were triple-marked to determine the loss rate of CWTs, freeze brands, and fin clips. Approximately 97,800 striped bass fingerlings were coded-wire tagged in fall 1986 at the California Department of Fish and Game's Central Valley Hatchery (CVH). The following spring, the 95,300 surviving coded-wire-tagged, age-1 fish (mean weight = 110 g) were freeze-branded with an "O" on the left side and the right pelvic fins were excised. These fish were stocked in the Sacramento River at Rio Vista (Fig. 1) on 1 May 1987, approximately 20 days after external marking.

In fall 1989, approximately 47,500 striped bass fingerlings were coded-wire tagged at CVH. The following spring, the 46,500 surviving coded-wire-tagged, age-1 fish (mean weight = 110 g) were freeze-branded with an "O" on the right side and the left pelvic fins were excised. These fish were stocked in San Pablo Bay at Rodeo (Fig. 1) on 15 May 1990, approximately 20 days after external marking.

Recovery

Marked fish are recovered in drift gill nets and fyke traps during the striped bass' spring spawning period and during a year-round creel census. These recovery efforts are part of another mark-recapture program initiated in 1969 designed to estimate abundance, mortality rates, and movement of legal-sized striped bass (Chadwick 1963, Stevens 1977, Stevens et al. 1985). Drift gill nets are fished in the lower reaches of the Sacramento and San Joaquin rivers from two boats. Fyke traps are fished in the Sacramento River near Clarksburg from 1981 to 1988, below Garcia Bend in 1989, and near Knights Landing from 1990 to the present. The creel census samples the Sacramento River and San Francisco Estuary, generally from Colusa to Pittsburg in late winter and spring and the bays and Pacific Ocean downstream of Rio Vista in summer and fall (Fig. 1).

Data from only 1 of the gill-netting boats were used to calculate loss rates because the electrical system of the 2nd gill-netting boat interfered with the operation of the CWT detector.

Personnel assigned to these recovery efforts were instructed to visually inspect each fish for the FB/FC marks and then check for presence of a CWT using a CWT detector. The left cheek muscle was removed from fish that elicited a positive response to the CWT detector and returned to the laboratory for dissection and CWT recovery.

[1]Use of trade names does not imply endorsement by the California Department of Fish and Game.

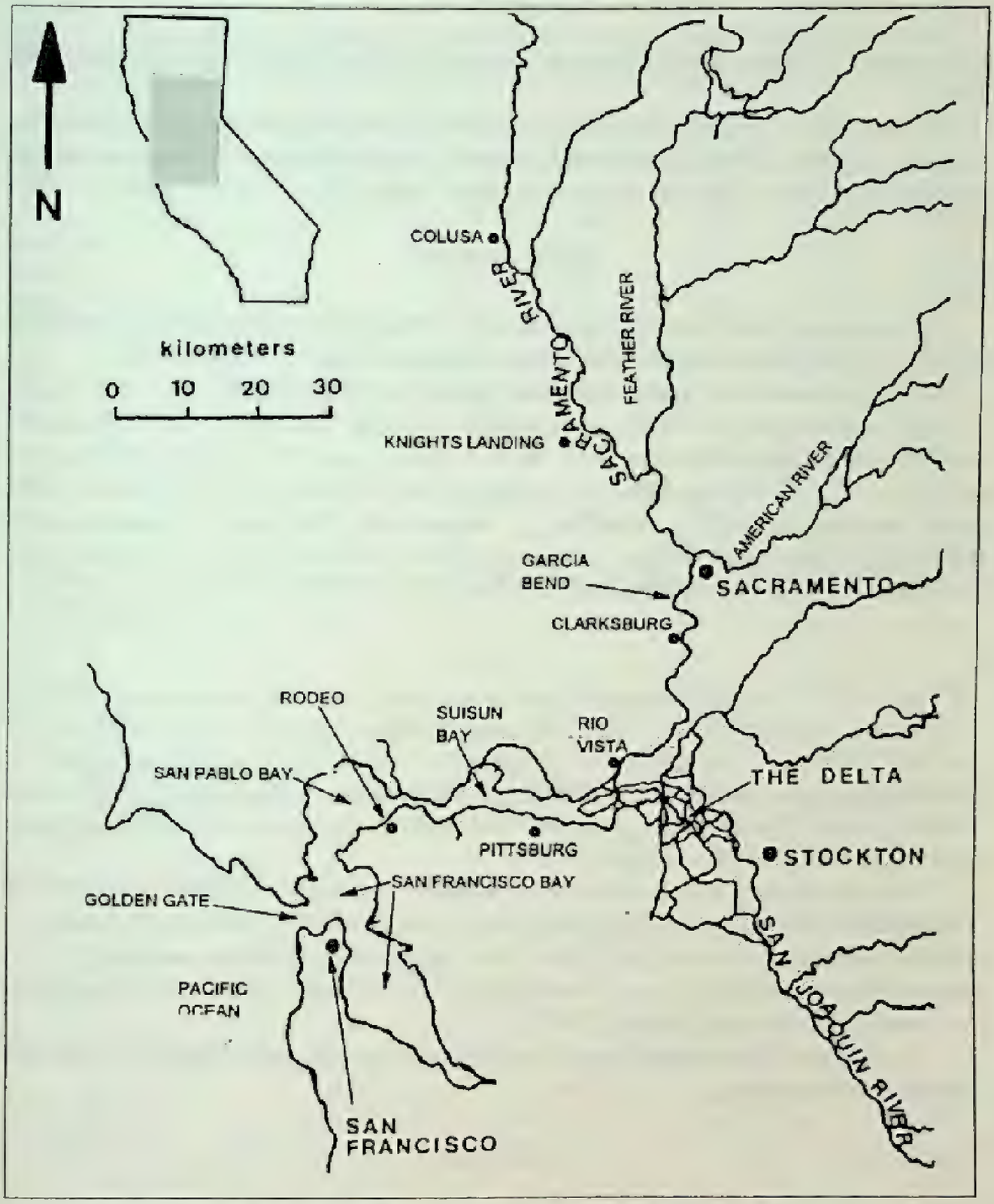

Figure 1. Map of the Sacramento-San Joaquin Estuary and upstream areas.

There were two different CWT detectors used during this study. Prior to 1990, a detection unit that required passing each fish through a 15-cm diameter, 76-cm-long tube (Northwest Marine Technology[1], Model FDA-84) was used to detect tagged fish. Only striped bass up to about 65 cm TL (4-5 years old) were able to fit through this tube.

Few marked fish from the 1986 year-class exceeded this length before 1990 (Collins 1982).

Hand-held, wand-style detectors (Northwest Marine Technology[1]) replaced the tube unit in 1990. The wand detector is more versatile than the tube detectors as it is simply passed across the left cheek of the fish, regardless of the fish's size.

Data Analysis

Although this study was designed as a triple-mark experiment, low recoveries of fin-clipped and freeze-branded fish led to combining them as a single mark.

Mark-loss rates were estimated using Myhre's (1966) equation (1), where "loss" includes the shedding of CWTs, regeneration of fin clips, fading of freeze brands, and non-detection of any of the marks. Loss of a mark, e.g., a CWT, is assumed to occur at the rate L_{CWT}. The proportion of a marked group of fish that loses or retains CWTs at the end of each year is L_{CWT} and $1\text{-}L_{CWT}$, respectively. The same notation applies to FB/FCs by replacing subscripts. Loss rate of CWTs is estimated as the proportion of recaptured double-marked fish that have lost their CWTs:

$$L_{CWT} = \frac{N_{FB/FC}}{N_{CWT,FB/FC} + N_{FB/FC}}$$

where $N_{CWT,FB/FC}$ is the number of double-marked fish recaptured with both a CWT and at least 1 external mark and $N_{FB/FC}$ is the number of double-marked fish recaptured with 1 or both external marks, but no CWT.

We used chi-square tests to determine if the overall loss rate of CWTs and FB/FCs differed between the 2 year-classes and if the change in CWT detectors in mid-1990 affected estimates of mark loss rates. For the latter comparison, we used a 3 x 2 contingency table where rows are "before 1990," "1990," and "after 1990" and columns are recoveries with and without CWTs.

Least squares linear regression is used to model the change in FB/FC loss with time for the 1986 year-class.

RESULTS

Through 1996, 520 double-marked fish were recovered from the 1986 year-class; no 1986 year-class fish were recovered after 1996 (Table 1). This was a recovery rate of 0.5%. Of these recoveries, 73.3% retained the CWT and at least 1 external mark, 14.4% retained only the CWT, and 12.3% had only an external mark(s).

The change in CWT detection equipment in 1990 did not significantly affect the estimates of CWT loss rates ($\chi^2 = 2.786$, df = 2, $P = 0.249$), although starting in 1990, the estimated loss rates were lower. The estimated loss rate of CWTs before 1990 was 16.7%, compared to 12.5% in 1990 and 12.3% after 1990 (Table 1). The estimated overall CWT loss rate for the 1986 year-class was 14.4%.

Table 1. Recoveries of double-marked, hatchery-reared striped bass from the 1986 year-class and annual mark loss rate estimates (L) in the Sacramento-San Joaquin Estuary.

Year	Age at recovery	CWT and FB/FC	CWT	FB/FC	Total	L_{CWT}	$L_{FB/FC}$
1988	2	27	6	8	41	0.229	0.182
1989	3	142	16	26	184	0.155	0.101
1988-89[a]						0.167	
1990	4	112	16	16	144	0.125	0.125
1991	5	69	18	7	94	0.092	0.207
1992	6	20	9	0	29	0	0.310
1993	7	7	9	5	21	0.417	0.563
1994	8	3	0	2	5	0.400	0
1995	9	0	0	0	0		
1996	10	1	1	0	2	0	0.500
1991-96[a]						0.123	
Total		381	75	64	520	0.144	0.165

[a]Estimates for the periods before and after 1990, when CWT detection equipment changed. See text for details.

The estimated overall FB/FC loss rate for the 1986 year-class was 16.5%. The FB/FC loss rate increased linearly with time, as indicated by the regression: loss rate = 0.1109(recovery year) – 220.54 ($r^2 = 0.87$) (Fig. 2). This figure depicts a FB/FC loss rate of 11% per year after starting at a loss of 4% at age 3 in 1989. In 1993, at age 7, total loss reached 48%.

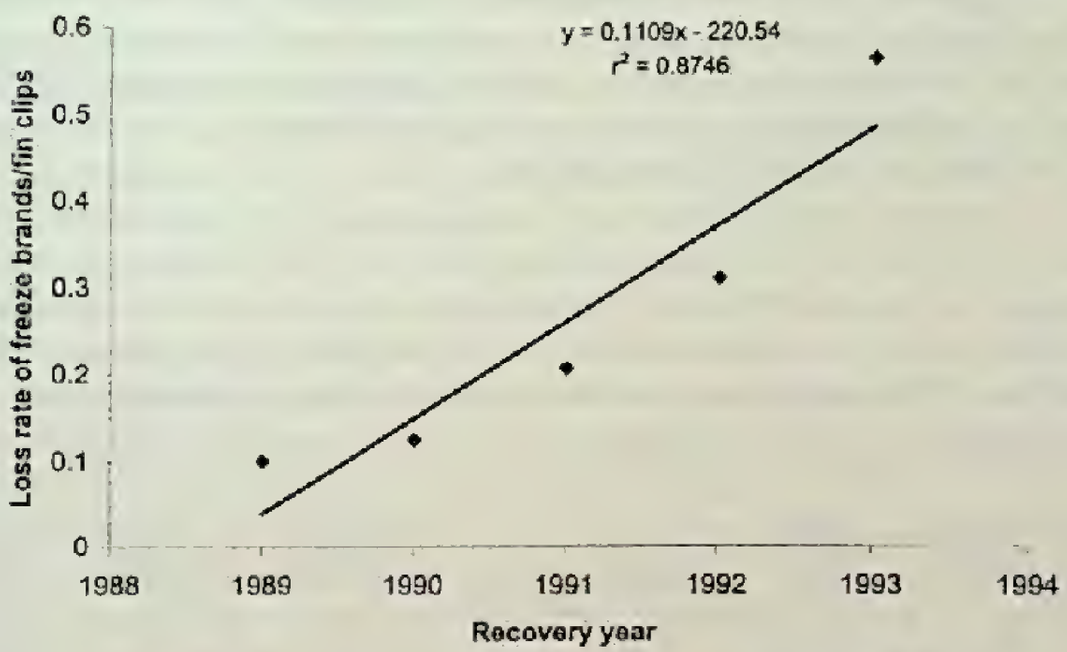

Figure 2. Annual loss rate of freeze brands/fin clips from hatchery-reared striped bass in the Sacramento-San Joaquin Estuary.

Only 93 (0.2%) double-marked fish from the 1989 year-class were recovered (Table 2). This was less than half the recovery rate of the 1986 year-class. Recoveries consisted of 38.7% with a CWT and FB/FC, 51.6% with CWT only, and 9.7% with FB/FC only. Similar to the 1986 year-class, no double-marked fish from the 1989 year-class were recovered after 1996. Estimated overall CWT and FB/FC loss rates were 20% and 57.1%, respectively (Table 2).

Table 2. Recoveries of double-marked, hatchery-reared striped bass from the 1989 year-class and annual mark loss rate estimates (L) in the Sacramento-San Joaquin Estuary.

Year	Age at recovery	CWT and FB/FC	FB/FC	CWT	Total	L_{CWT}	$L_{FB/FC}$
1991	2	1	1	5	7	0.500	0.833
1992	3	16	1	17	34	0.059	0.515
1993	4	15	3	23	41	0.167	0.605
1994	5	4	4	2	10	0.500	0.333
1995	6	0	0	0	0		
1996	7	0	0	1	1	0	1
Total		36	9	48	93	0.200	0.571

The CWT loss rates were not significantly different between the two year-classes ($\chi^2 = 0.520$, df = 1, $P = 0.471$), but the FB/FC loss rates did differ ($\chi^2 = 68.029$, df = 1, $P < 0.001$) (Table 3). Because the difference in estimated FB/FC loss rates might be due to human error in detecting external marks, we compared the 1986 and 1989 year-class recoveries for 1991–1996 only, a period when field samplers was the same. The FB/FC loss rates for this time period were also significantly different between the two year-classes ($\chi^2 = 19.98$, df = 1, $P < 0.001$) (Table 4).

Table 3. Retention and loss of coded-wire tags (CWT) and freeze brands/fin clips (FB/FC) for the 1986 and 1989 year-classes of hatchery fish recovered in the Sacramento-San Joaquin Estuary.

Year-class	Retained CWT	Lost CWT	Total	Retained FB/FC	Lost FB/FC	Total
1986	456	64	520	445	75	520
1989	84	9	93	45	48	93
Total	540	73	613	490	123	613

Table 4. Recoveries in 1991-1996 of coded-wire tags (CWT) and freeze brands/fin clips (FB/FC) in the Sacramento-San Joaquin Estuary for the 1986 and 1989 year-classes when field samplers were the same.

Year class	CWT+FB/FC	CWT	Total
1986	100	37	137
1989	36	48	84
Total	136	85	221

DISCUSSION

The estimated CWT loss rate in this study was surprisingly high, especially in comparison with those observed for captive striped bass in shorter-term experiments (Klar and Parker 1986, Wooley et al. 1990, Wallin and Van Den Avyle 1994). However, previous studies were not done in the wild and detection of tags was not subject to the vagaries inherent in field sampling programs.

We have found no reports describing loss of freeze brands or fin clips on striped bass. Experiments with other species are consistent with our results in that recognition of these external marks decreases with time (McNeil and Crossman 1979, Haines and Modde 1996, Conover and Sheenan 1999).

Besides fading of freeze brands and regeneration of fins, factors that may affect detection of marks include weather conditions, skill and motivation of the field samplers, and degree of angler cooperation. Inclement weather and unfavorable water conditions on the gill-netting boats affect the ability to detect FB/FCs and degrade the reliability of the CWT detectors, especially the tube detectors used before 1990. Lighting conditions (bright sun, overcast, glare from the water) affect the ability to detect FB/FCs, especially freeze brands. In spite of extensive training, field samplers, who are generally temporary employees, vary substantially in their skill and motivation. Bias from this source may be particularly great if samplers search more diligently for 1 mark when another mark is detected first. Finally, angler cooperation is crucial during the creel census and the lack of it can result in rushed and incomplete sampling.

Detection of CWTs may be influenced by the reliability and versatility of the detector. Although the estimated loss rate of CWTs was not statistically different due to the change in detector types in 1990, other factors warranted use of the wand detector in preference to the tube unit. The tube unit appeared less reliable, giving false positive signals when live fish "flopped" as they passed through the tube or when other motion jarred the detector. The latter often occurred on sampling vessels and was compensated by reducing the sensitivity of the detector, sometimes resulting in the failure to recognize tagged fish. Although false positive errors could be reconciled in the laboratory when the CWT was extracted, there was no remedy for fish that gave false negative readings.

Other studies of CWT retention led us to expect that most tag loss would occur

during the period between fall tagging and stocking the following spring. Before stocking, a quality control check was conducted to determine the fraction of the fish retaining CWTs and the number of tagged fish was corrected accordingly. We did not expect substantial tag losses after release. These high loss rates are likely attributed to non-detection rather than to actual shedding of CWTs.

The differences in results between the double-marking experiments in 1986 and 1989 are unexplained. The 1989 year-class recovery rate was less than half the recovery rate of the 1986 year-class, suggesting lower survival or, in the case of FB/FCs, poorer quality control. Previous observations of freeze-branded and fin-clipped striped bass from the 1981 and 1982 year-classes maintained at the California Academy of Science's Steinhart Aquarium suggest that the quality of fin clips and freeze brands may vary among years. Fish from the 1981 year-class were inspected in 1983 and 90% of them had discernable brands and identifiable clips. Conversely, only about half of the 1982 year-class had identifiable fin clips after only 3 months (California Department of Fish and Game, unpublished data).

The low number of recoveries from the 1989 year-class and the inconsistencies in the loss estimates for that year's experiment (Table 2) suggest that only estimates from the 1986 experiment should be used to correct marked-fish recoveries for mark loss and non-detection. Even though there was no significant difference in CWT loss rates before and after 1990, our assessment of the unreliability of the tube detectors used before 1990 suggests that use of different loss rates in these 2 periods is appropriate. Consequently, CWT recoveries before 1990, when tube detectors were used, will be corrected using a loss and non-detection value of 16.7%. Starting in 1990, CWT recoveries will be corrected using a loss and non-detection value of 12.3%. Corrections for FB/FC losses will be based on the 1986 year-class regression model that yields estimated loss of FB/FCs of 4% at age 3, 15% at age 4, 26% at age 5, 37% at age 6, and 48% at age 7.

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DIET OF BULLFROGS IN RELATION TO PREDATION ON GIANT GARTER SNAKES AT COLUSA NATIONAL WILDLIFE REFUGE

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To evaluate predation by bullfrogs, *Rana catesbeiana*, on giant garter snakes, *Thamnophis gigas*, at Colusa National Wildlife Refuge (NWR) we collected 99 bullfrogs during three field seasons from 2000-2002. Crayfish, *Procambarus clarkii*, were in 90% of the bullfrogs and were usually the only food item. We found four neonate giant garter snakes in three bullfrogs. We estimated the total annual predation of bullfrogs to be about 22% of neonate production, a value similar to the 20% tail loss we documented for giant garter snakes on the Refuge as another index of bullfrog predation pressure. Even with mortality from bullfrogs and other predators, snake size classes indicate sustainable recruitment into the Colusa NWR giant garter snake population. Smaller bullfrogs and bullfrog tadpoles are also food for giant garter snakes, so further studies are needed to determine the net effect of bullfrog removal for local giant garter snake populations.

INTRODUCTION

Bullfrogs, *Rana catesbeiana*, prey on a wide variety of invertebrates and small vertebrates (Bury and Wheland 1984). Predation pressure from non-native bullfrogs have been implicated in the decline of native frogs in California (Fellers and Drost 1993, Hayes and Jennings 1986) and native frogs and garter snakes in Arizona (Rosen and Schwalbe 1995). Investigators (Crayon 1998, King et al. 2002) have recently recorded bullfrogs eating a Pacific Coast aquatic garter snake, *Thamnophis atratus hydrophilus*, and a Sonoran gopher snake, *Pituophis catenifer affinis*. As part of our current study of giant garter snakes, *Thamnophis gigas*, at Colusa National Wildlife Refuge (NWR),

Colusa County, CA, (Wylie et al.[1,2] 2000, 2002), we were interested to what extent the abundant non-native bullfrogs on the Refuge preyed on these endemic aquatic snakes, which are federally and state listed as a threatened species. Few quantitative studies have been conducted on predation of wild populations of snakes. Predation on young water snakes is thought to be a significant source of mortality for snake populations (Mushinsky and Miller 1993) and bullfrogs have been shown (Willis et al. 1982, Schwalbe and Rosen 1988) to be a significant source of predation on Mexican garter snakes, *Thamnophis eques*. Identifying sources of mortality for giant garter snakes would be useful in managing the recovery for this listed species. Bullfrogs are abundant on the Colusa NWR, but an evaluation on their effects on native amphibians and reptiles on the Refuge has yet to be determined. We reported the early results of this study (Carpenter et al. 2002), which confirmed that bullfrogs did eat young-of-the-year giant garter snakes at Colusa NWR. The purpose of this paper is to report on the diet of bullfrogs sampled over three field seasons and to examine their potential predation pressure on giant garter snakes on the Refuge.

METHODS

We conducted this study on Colusa NWR located 2 km west of the town of Colusa in Colusa County. The Refuge comprises 1872 ha and we conducted our work on the southern third of the Refuge. During three field seasons (2000-2002) we captured giant garter snakes in modified floating minnow traps (Casazza et al. 2000) and by hand. We measured each snake for snout to vent length (S-VL) and vent to tail tip length (V-TTL). We made notation of tail loss and used a ratio of VTTL:SVL <0.145 to signify appreciable tail loss likely due to predator attacks (Willis et al. 1982). During 2000-2002 we captured bullfrogs in late July to early September, when giant garter snakes usually give birth to their young (Hansen and Hansen 1990, Wylie[3] 1997). We collected bullfrogs in areas in which we caught giant garter snakes. We caught frogs at night using spotlights and gigs. We sampled 30 frogs in 2000, 39 in 2001 and 30 frogs in 2002 for a total of 99 frogs. We pithed bullfrogs immediately after gigging and then froze them after we determined their sex, weight, and snout-vent length. We later examined stomach contents from thawed specimens. We identified stomach contents to the lowest taxonomic level their condition would allow.

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[1]Wylie, G.D., M.L. Casazza, and N.M. Carpenter. 2000. Monitoring giant garter snakes at Colusa National Wildlife Refuge: 2000 progress report. U.S. Geological Survey, Dixon, California. 14 pp.

[2]Wylie, G.D., M.L. Casazza, and N.M. Carpenter. 2002. Monitoring giant garter snakes at Colusa National Wildlife Refuge: 2001 progress report. U.S. Geological Survey, Dixon, CA. 13 pp.

[3]Wylie, G.D., M.L. Casazza, and J.K. Daugherty. 1997. 1996 Progress report for the giant garter snake study. U.S. Geological Survey, Dixon Field Station, Dixon, CA. 15 pp.

RESULTS AND DISCUSSION

Bullfrogs we collected ranged from 130-198 mm SVL and 140-690 g, with females slightly bigger than males (Table 1). Twenty-one invertebrate and four vertebrate taxa were represented in bullfrog stomach contents (Table 2). Only 4 of 99 bullfrogs had empty stomachs. The diversity of prey items in these bullfrogs reflects the wide range of diet reported in the literature (e.g., Bury and Whelan 1984, Schwalbe and Rosen 1988, Cross and Gerstenberger 2002). Of the bullfrogs in our study, however, 90% had one to four crayfish, *Procambarus clarkii*, in their stomachs with other prey items largely incidental (Table 2). These crayfish are also an introduced species to California. Crayfish were also the principal food items for bullfrogs in various lotic and lentic habitats in Missouri (Korschgen and Baskett (1963) and were preferred over fish in goldfish culture ponds (Corse and Metter 1980). The abundance of these non-native crayfish at Colusa NWR may be key to supporting the large numbers of non-native bullfrogs on the Refuge.

Table 1. Statistical summary of length (mm) and weight (g) of bullfrogs captured during mid-July to early September 2000-2002 at Colusa National Wildlife Refuge.

	Female Length	Female Mass	Male Length	Male Mass
Mean	162	386	156	357
Median	162	354	155	360
Minimum	118	140	130	232
Maximum	198	690	180	600
SE	19.02	131.2	11.9	74.8
N	44	44	55	55

We found the remains of three neonate giant garter snakes in the stomachs of two bullfrogs from our 2000 field season and the remains of one snake in one bullfrog form our 2002 field season. No giant garter snakes were found in bullfrogs during 2001. The four giant garter snakes were found in samples taken during July (one) and August (three). Bullfrogs containing giant garter snakes were also among the smaller to medium-sized individuals we sampled (270-360 g), which indicates the wide size range of bullfrogs capable of eating giant garter snakes. Bullfrogs have been shown to eat much larger snakes (e.g., Crayon 1998, Rosen and Schwalbe 1998), but we found no sub-adult or small male snakes in the bullfrogs we sampled. Perhaps behavior of larger giant garter snakes or structural complexity of their habitat allows them to avoid predation by bullfrogs even though they share much of the same habitat. Bullfrog predation may be more severe on snakes in habitat with little structural complexity.

Of the giant garter snakes we captured during this study, 53 of 265 had suffered appreciable tail loss. This 20% tail loss is greater than that of 10-13% for populations of three *Thamnopis* species in Michigan (Willis et al. 1982), but is less than 62% for *Thamnophis eques* in Arizona (Schwalbe and Rosen 1988). If bullfrogs are the source

Table 2. Stomach contents from 99 bullfrogs captured during mid-July to early September 2000-2002 at Colusa National Wildlife Refuge.

Food Items	Males (N=55)		Females (N=44)	
	N	No. Frogs	N	No. Frogs
Arachnida				
Araneae	3	2	0	0
Amphibia				
Rana catesbeiana (tadpole)	0	0	1	1
Chilopoda	0	0	1	1
Coleoptera				
Corixidae	0	0	1	1
Dytiscidae	7	6	4	3
Hydrophilidae	0	0	1	1
Notonectidae	2	2	0	0
Scarabidae	1	1	3	2
Coccinellidae	1	1	0	0
Crustacea				
Isopoda	1	1	0	0
Decapoda				
Procambarus clarkii	85	48	50	41
Hemiptera				
Pentatomidae	1	1	0	0
Hymenoptera				
Vespidae	0	0	8	3
Apidae	2	2	0	0
Lepidoptera				
Unknown moth	1	1	0	0
Unknown caterpillar	1	1	1	1
Oligochaeta	1	1	0	0
Odonata				
Anisoptera	1	1	1	1
Zygoptera	1	1	1	1
Orthoptera				
Acrididae	1	1	0	0
Gryllidae	2	2	0	0
Mantidae	1	1	0	0
Stenopelmatidae	3	1	0	0
Pices				
Gambusia affinis	0	0	3	3
Rodentia				
Microtus	0	0	1	1
Serpentes				
Thamnophis gigas	3	2	1	1

of most of this tail loss (Schwalbe and Rosen 1988), tail loss could be used as an indicator of bullfrog predation pressure. Giant garter snakes have coexisted with bullfrogs at Colusa NWR for many years, and bullfrog predation has not severely limited recruitment of young snakes into the population, as is the case for Mexican garter snakes in Arizona (Schwalbe and Rosen 1988). The size structure of the giant garter snakes we captured shows a good representation of medium to large individuals with fewer snakes represented in the smaller size classes (Fig. 1). Mortality rates from all sources are likely higher for smaller, younger individuals and decrease with age and size (Willis et al. 1982). In addition, catching snakes by hand and with traps likely does not represent the true abundance of the smallest giant garter snakes. Overall, the population structure depicted in Figure 1 suggests sustainable recruitment of smaller snakes into the giant garter snake population at Colusa NWR.

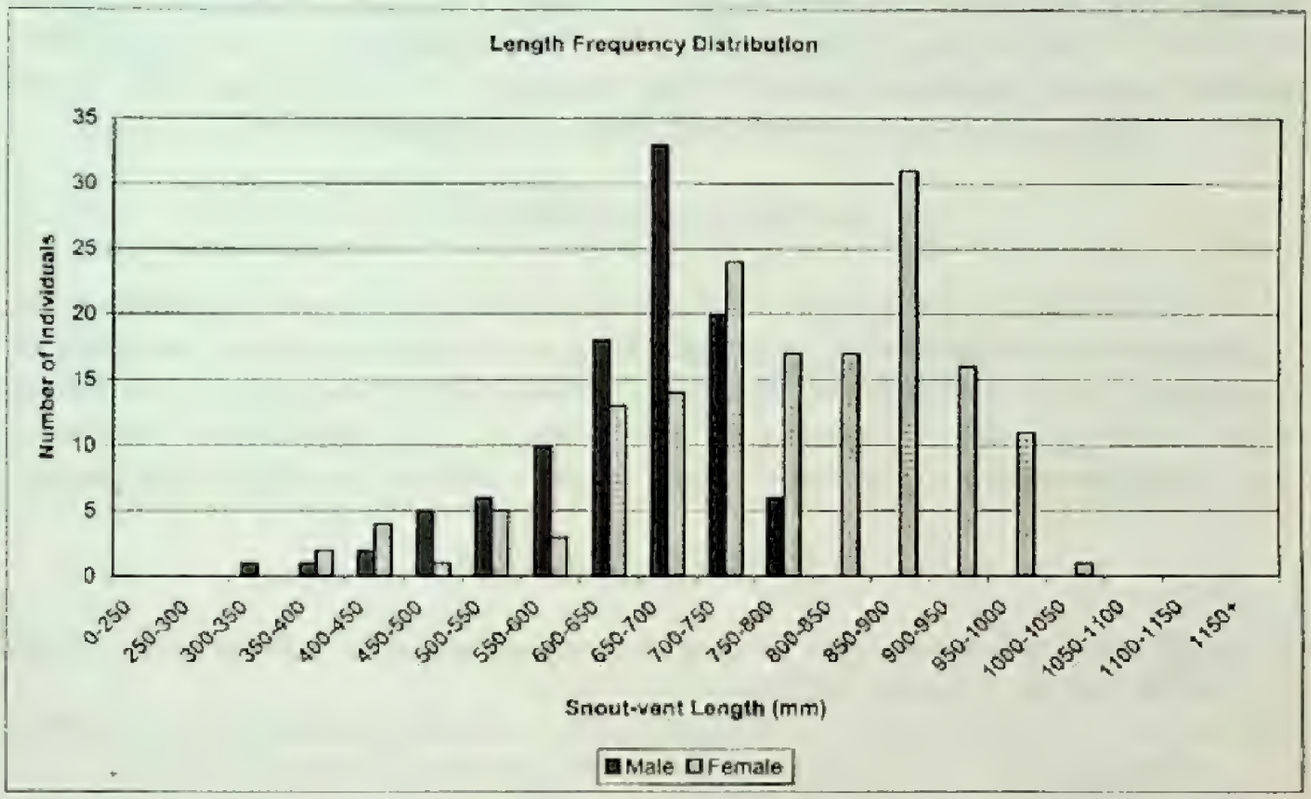

Figure 1. Size classes of 265 giant garter snakes captured during 2000-2002 at Colusa National Wildlife Refuge.

Our study confirms bullfrogs as predators of giant garter snakes. The additive impact of their predation on giant garter snake populations is still unknown, but we can model bullfrog predation based on several assumptions. We feel we have sampled no more than 10% of the bullfrogs in our study area, which would yield an estimate of about 330 bullfrogs present in any given year. Predation rate on giant garter snakes is about 4% based on stomach contents, which we assume are representative of two feeding days. Our field observations indicate giant garter snakes have their young over an approximate 45-day period. From our capture data for giant garter snakes we estimate

about 60 reproductive females are present in any given year. Litter size of giant garter snakes has been estimated at 23 (Cunningham 1959, Hansen and Hansen 1990), so the total reproduction in any year is about 1380 neonates. Using the above estimates, a predation rate of 0.04/2 days times 330 bullfrogs multiplied by the 45-day reproductive period yields a loss of 297 neonate giant garter snakes from 1380 neonates born. These assumptions yield an annual mortality of 21.5% of neonate giant garter snakes from bullfrog predation, which is close to the 20% rate of tail loss we calculated.

A large population of giant garter snakes is maintained in diverse wetland habitat on Colusa NWR even with bullfrogs and other predators, such as wading birds, raptors, and otters. The abundant crayfish population at Colusa NWR is a major food supply for bullfrogs and likely for these other predators. The overall impact of bullfrog predation on the giant garter snake population at Colusa NWR is difficult to estimate because these snakes also prey on tadpoles and small bullfrogs (Fitch 1941, Rossman et al. 1996, Wylie[3], 1997), and the benefits of bullfrogs as food may mitigate some of their costs to these snakes as predators. Further studies are needed to determine the net effect of bullfrog removal for local populations of giant garter snakes.

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ASSIMILATION OF PELAGIC RED CRABS, *PLEURONCODES PLANIPES*, BY WESTERN GULLS, *LARUS OCCIDENTALIS*.

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Western gulls, *Larus occidentalis*, ate exclusively pelagic red crabs, (*Pleuroncodes planipes*) during a mass stranding. The gulls regurgitated and excreted the crustaceans at specific site in San Benito Islands, allowing collection of fresh samples. A total 347 live red crabs, 63 regurgitated, and 86 scats was dried, frozen, and analyzed for proximate composition. Nutritional value of crabs was measured for its assimilation. Whole crabs had 9.46% of lipids and 40.14% of crude protein, regurgitated crabs 0.29% and 16.31%, while feces contained 0.06% and 24.96%, respectively. Assimilation for lipids and protein in regurgitated crabs was 96.9% and 59.3%; feces indicated 99.9% lipid assimilation but crude protein enrichment, probably due to bacterial load. Considering lipid fraction, regurgitation seemed more efficient since most of lipids were assimilated with a partial and less costly digestion. Western gulls prey on *P. planipes* not only because its abundance or easier catch, but also because of its nutritional value when regurgitated.

INTRODUCTION

Between the 8th and 18th of February 2001, pelagic red crab swarms, *Pleuroncodes planipes*, surfaced near the Islas San Benito, Baja California, México (28° 18' N; 115° 113' W). Proximity of the crabs to the coast, combined with strong winds and receding tides, resulted in a mass stranding on beaches along the narrow channel that separates the West and Middle islands (Fig. 1). When red crabs are abundant at the sea surface or become stranded, many species of fish and sea birds prey intensively on the crustaceans (Stewart et al. 1984; Balart and Castro 1995). Several gulls species such as Herring gull, *Larus argentatus*, ringed-bill gull, *Larus delawarensis*, and Heerman's gull, *Larus heermanni*, feed on red crabs during El Niño events when crustaceans move into California waters as far north as Monterey, California (Glynn 1961; Stewart et al. 1984). Because of the red crabs easy capture and abundance, the gulls fed intensively on them during these events (Stewart et al. 1984). In the present study, we focus on evaluating the nutritional benefit of red crabs to gulls under such special field conditions.

At the time, a colony of about 300 western gulls, *Larus occidentalis*, fed on the red crabs. The gulls were nesting in a valley on Middle Island about 600 m from the feeding area. This proximity of nesting and foraging sites enabled us to collect live red crabs

Figure 1. Location of San Benito Islands off the west coast of Baja California. The inset map indicate the location of crab strandings and gull nesting areas.

at the stranding site, fresh spewings (regurgitated pellets), and scats at the nesting grounds. The gulls fed exclusively on red crabs as reflected by the pellets, which contained solely red crabs, and the scats, which were bright red and contained fragments of exoskeleton. These related items allowed us to evaluate the assimilation

efficiency of free-ranging gulls feeding exclusively on this crustacean.

Some proximate composition analyses exist for red crabs in their different stages, (pelagic, benthic-pelagic and strictly benthic) latitudes and seasons off Baja California (Aurioles et al. 1994; Castro et al. 1995). Carrillo-Domínguez, et al. (1995) tested red crabs as a component of poultry food and found that it was an excellent additive.

On February 10, we collected 347 stranded live red crabs while they were still very active. The individuals were sexed and the cephalothorax length (mm) was measured. Most of the females examined had attached eggs. The sex ratio was 1: 1.13 (females:males). The red crabs were sun-dried and stored in a plastic bag to later be frozen. This procedure was chosen to avoid decomposition during the period where refrigeration was not available (plane and truck transportation).

Red crabs ranged from 14 to 25 mm long with a mode 20-21 mm interval. This size distribution was similar to another obtained from a Bahía Magdalena stranding occurred in May 1991 (12-26 mm), from which a proximate analyses was also conducted (Aurioles et al. 1994). The size of the crabs in both strandings corresponded to individuals approaching their first year of age, still in the pelagic phase. These crustaceans were breeding for the first time, which usually do during the months from December through March (Boyd 1967; Aurioles 1992). During this period of the year the pelagic red crab is very abundant in the area from 24° to 28° N., off West coast of Baja California (Aurioles 1992).

Within 24 hours, we collected 63 regurgitated pellets and 86 scats of western gulls at the nesting area. Because they were scattered over a large area, we assumed the scats came from many different individuals. The pellets and scats were also sun-dried and frozen during the same period as the whole red crabs.

In the laboratory, all types of samples were homogenized, dried, and duplicate proximate analyses were made for each type of sample (red crabs, regurgitated pellets, and feces). The techniques used were those of the Association of Official Analytical Chemists (A.O.A.C., 1984); moisture (7.007), ash (7.009), crude fibre (7.060), carbohydrates (by difference). The composition for the stranded red crabs (Table 1) was similar to the composition determined in another stranding (May 1991) in Bahía Magdalena B. C. S. (Aurioles et al. 1994), except for the ether extract and carbohydrates less abundant in this study.

When the proximate compositions between stranded (expected composition) and regurgitated pellets (observed composition) are compared (Table 1), the proportions were significantly different (x^2, test, $p > 0.05$). Relative to whole crabs, moisture and ash content were greater in the pellets, while 97.2% of the lipids (ether extract) and 64.3% of the crude protein (protein ingested minus protein regurgitated/protein ingested x 100), were assimilated by the gulls. In the scats, the relation between ingested and excreted crude protein (Table 1) indicated an assimilation of 41.14%, a smaller value than for the regurgitated material (64.3%). This difference may possibly be attributed to a much higher bacterial load in feces. Also, as protein and lipids are assimilated, crude fiber increased its percentage in both the regurgitated and excreted samples.

The assimilation efficiency of the lipid fraction was 99.38%, based upon the composition of the scats, only 2.18% higher than for the regurgitated material (97.2%).

The difference in lipid content between regurgitated and excreted red crab was not significant (Table 1). The data suggests that, for high energy, it would be more convenient to ingest and regurgitate the red crabs, instead of completing the digestion, because the small difference of assimilation between the regurgitation and excretion for the lipid fraction. In other words, most of the lipids are readily assimilated in the regurgitation and any further digestion will extract just a minor percentage with probably a larger use of energy to obtain it. Additionally and because the red crab has a high concentration of ash (minerals) (25.31%), the complete digestion process to

Table 1. Mean values for proximate chemical composition of stranded pelagic red crabs, regurgitated and excreted by Western gulls *Larus occidentalis*.

	Moisture	Ash	Lipids	Crude protein	Crude fiber	Carbohydrates
Condition						
Stranded (Benitos)						
Mean	14.14	25.31	8.12	34.46	5.97	11.97
S. D.	0.043	0.78	0.60	1.35	0.17	
Regurgitated						
Mean	24.59	42.31	0.22	12.30	8.66	11.89
S. D.	0.009	1.09	0.05	0.15	0.18	
Assimilation			97.29	64.30		
Difference	+	+	+	+	+	-
Excreted						
Mean	18.73	48.44	0.05	20.28	9.75	2.73
S. D.	0.08	0.33	0.001	0.35	0.009	
Assimilation			99.38	41.14		
Difference	+	+	+	+	+	+
Difference Regurgitated *versus* Excreted	+	+	-	+	-	+
Stranded Bahía Magdalena*						
Mean	4.71	29.04	14.12	41.19	8.30	2.65
S. D.	0.20	0.24	1.98	0.22	0.62	

* Aurioles et al. 1994.
All comparisons between stranded, regurgitated and excreted based on X^2 test:
+ = significant difference; - = no difference (Alpha = 0.05).

separate the nutrients from the organic matter would be likely a more costly process compared to the partial digestion and regurgitation. Similar behavior is common in owls that regurgitate pellets containing indigestible material left in the gizzard (teeth, skulls, claws, and feathers), that otherwise might be dangerous to pass through the rest of the digestive tract.

This finding predicts that regurgitated pellets should be more abundant (in biomass) than scats. This information was, unfortunately, not recorded in the field, but it would be possible in future studies of red crab strandings.

The availability of red crabs as food for western gulls and other seabirds is limited to those occasions when the crustaceans swarm at the surface and/or become stranded. The gulls use of crabs for food is probably not determined solely by the crabs' easy capture and abundance when swimming or stranding, but also because they are an important nutritional item that might increase or sustain gull productivity when other resources are not available, as documented during El Niño events.

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NEW HOST RECORD OF AVIAN TUBERCULOSIS IN AN AMERICAN WHITE PELICAN, *PELECANUS ERYTHRORHYNCHOS*

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An American White Pelican, *Pelecanus erythrorhyncos,* (AWPE) was captured from an airboat on August 9, 1999, in the Wister Waterfowl Management Unit on the Salton Sea in Imperial County, CA. The U. S. Fish and Wildlife Service and California Department of Fish and Game conduct routine monitoring of the Sea's shorelines throughout the year to document disease outbreaks and to collect sick and dead birds. This monitoring program was initiated in response to a large outbreak of type C botulism in fish-eating birds in 1996. Over 15,000 birds, including nearly 9,000 American white pelicans were killed during this disease event (Rocke et al. 2003, in press). Since then, smaller outbreaks have occurred every summer. From June through October, pick-ups predominantly consist of pelicans, both AWPE and California brown pelicans, *Pelecanus occidentalis californicus.* In this case, the AWPE was observed in the water, unable to fly away or evade the boat. The bird was in good condition otherwise and was brought back to a field hospital at the Sonny Bono Salton Sea National Wildlife Refuge for further observation. During the night, the bird was able to walk upright and showed considerable aggression towards staff. Since it did not display typical signs of avian botulism and seemed strong and alert, the bird was banded and released onto the Sea the following morning. On August 11, the same pelican, identified by the band, was found in the release area profoundly weaker and minimally responsive. The bird was subsequently euthanized with carbon dioxide and necropsied. The animal was in good body condition, having adequate internal fat and musculature. Numerous multifocal, 1-mm diameter, yellow masses were observed throughout the lungs and air sacs. The liver was yellow. A 6-cm diameter, firm spherical mass was present in the abdominal cavity, distal to the liver. This mass was suspected, and later determined, to consist of splenic tissue. Intestinal contents consisted of green-brown, thin fluid. No other lesions were noted.

Tissues (10% formalin-fixed and frozen) were sent to the USGS/National Wildlife Health Center (NWHC) for further diagnostics. Histopathological examination was performed on formalin-fixed spleen, lung, liver and colorectum. Normal splenic

architecture was replaced by multiple to coalescing granulomas. Each granuloma was composed of a central necrotic core surrounded by broad zones of chronic inflammatory cells consisting primarily of macrophages with widely scattered heterophils and isolated multinucleated giant cells. Separating and interspersing individual granulomas were moderate bands of fibrosis. Within necrotic regions were numerous small rod-shaped bacteria. Multinucleated giant cells contained "ghosts" of rod-shaped bacteria. Similar giant cell granulomas were present in lung, liver, and colorectum.

Acid fast bacteria were cultured from frozen spleen, lung, and air sac tissues. *Aspergillus* spp., including *A. niger*, were also cultured from the air sac tissue. Heart blood tested negative for type C botulinum toxin by mouse protection assay. *Mycobacterium avium* was identified from the splenic mass by the National Veterinary Services Laboratory (NVSL) in Ames, Iowa. This is the first reported case of avian tuberculosis in an American white pelican and was the first submission of a pelican with *M. avium* to the National Wildlife Health Center. *M. avium* has been identified in a number of free-ranging avian species, including ducks, swans, gulls, cranes and raptors (Gale 1971). The incidence of avian tuberculosis in free-ranging birds is generally estimated to be less than 1% (Smit et al. 1987; Friend 1999), though the prevalence of infection among wild waterfowl in North America has not been determined (Wobeser 1997). Captive bird populations seem to experience higher prevalence of disease, often being associated with stress, overcrowding and/or immune insufficiency. Non-avian species that are susceptible to infection include rabbits, mink, deer, pigs and other livestock, dogs, cats, and humans (Gerlach 1994). It is unknown if the pelican in this report was experiencing a primary stress or disease process that may have made it susceptible to both *Mycobacterium* and *Aspergillus* infection. In addition, this pelican may have been experiencing concurrent effects of type C botulism, despite the negative results of the mouse bioassay.

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